Support Document #13

February 23, 1995 Draft

Mammalian Risk Assessments

Introduction

Consistency questions have been raised in relation to the approach the branch uses to evaluate impacts to mammalian species. A cursory review of several RED's showed that, in general, approaches used were similar with major differences in how the information was being presented. However, the differences, in most instances, would not lead to different conclusions on potential hazard to wild mammals. However in an effort to limit potential perceived inconsistencies of our assessments, I've been asked to put together the following outline for assessing impacts to non-target mammalian species from the use of pesticides. As with all assessment procedures we use, they must be adapted to the particular circumstance being evaluated and modifications made where necessary.

In a couple of instances in the RED's, mammals were not addressed, as has been the case in reviews of other actions done in the branch. This absence of a mammal section seems to cause extreme consternation by some not realizing that for most anticholinesterase inhibitors, the avian assessment is protective of mammalian species. As most of us are aware, birds and mammals respond similarly to xenobiotics with any differences more quantitative than qualitative, with birds having lower hepatic microsomal mono-oxygenase and A-esterase activity than do mammals which makes birds more susceptible to both OPs and CBs in general. Hence, our concentration on avian species, if safe to birds, safe to mammals. This approach would seem to be still valid, however, for clarity all evaluations should include a section on mammals, at least stating the difference in toxicity values and pointing out that conclusions for avian species are appropriate for mammals. In cases where potential impacts to avian species are identified, then further consideration of mammal species seems germane to provide a more complete picture of potential impacts to non-targets.

Mammalian Toxicity Information

In most cases, particularly for new chemicals, mammalian toxicity data is from the human and domestic animal data base required for registration under subdivision F. For older chemicals some of the various compendiums will have additional values, sometimes on other mammalian species than those typically used for the human toxicity tests. When available, these should be considered along with the human tox data.

The Human tox data base consists of numerous tests of which four are the most pertinent to our initial screen for evaluating a chemical:

> Acute oral toxicity test, usually with the laboratory rat;
>
> Sub-chronic oral toxicity: 90-day study usually with the laboratory rat. Animals can receive the test substance in their diet, dissolved in drinking water, capsules, or gavage.;
>
> Chronic toxicity test: 12 or 24 months depending on food or non food use; routes of exposure can be oral, dermal, or inhalation, with oral preferred. Animals can receive the test substance in their diet, dissolved in drinking water, or gavage.
>
> Two generation reproductive test: Approximately 37 week study with test substance administered in the diet or water; however, gavage or capsule is an option.

The other tests, dermal, inhalation, etc. also need to be considered and used where appropriate.

EXPOSURE ESTIMATES

Non-granular Formulations

Mammal exposure estimates for non-granular formulations should be similar to those we use for avian species: EECs from "Kenaga". However, in a recent evaluation of the Kenaga nomogram by Fletcher et al.(1994) some minor modifications were recommended. This study reexamined the Kenaga nomogram using voluminous information complied at the University of Oklahoma in the UTAB database. This database has 42,000 individual records pertaining to over 1,000 different organic chemicals, 65% of which are pesticides. These data were compiled from the review of over 2,100 published papers. There are over 400 species of plants in UTAB, Representing 95 plant families and all major crops. In this study comparison of values in the UTAB data set to values predicted by the Kenaga nomogram showed the existing nomogram is, in general, accurate; however, there are some exceptions.

The residue levels for each of the six categories in the current nomogram are based on the highest value reported for each category in the survey published by Hoerger and Kenaga. The levels for four of these categories, short range grass, long range grass, leafy crops, and pods, appear to be satisfactory based on this study. They found very few occasions when values reported in the literature for these categories exceeded the nomogram predictions. In contrast, the residue levels predicted for forage crops and fruit were often lower than those reported in the literature and

were believed in need of revision. They suggested that based on their study the current nomogram values for forage should be increased from 58 ppm to 135 ppm and for fruit from 7 ppm to 15 ppm; then the number of exceeding values on day 0 would be reduced from 21 to 5 (22.1 to 5.2%) for forage crops and from 21 to 9 (19.4 to 8.3%) for fruit.

They further suggested merging categories with similar values. They suggested that since the 135 ppm for forage crops is close to the existing value of 125 ppm for leaf crops, it would be appropriate to combine the two into a broad leaf category and set the prediction value at 135 ppm. In a similar fashion, fruits (new value 15) and pods (old value 12) could be placed in a single category set at 15. Implementing these suggestions, as recommended here, reduces the number of categories from six to four with values as shown in table one. Using these revised estimates residue levels on the various wildlife food sources can be calculated by the following equation:

ppm on food source = use rate (lbs ai/A) X estimate residues

This study did not address insect residues. However, the approach utilized to estimate insect residues in the past proposed by Kenaga (1973) still appears appropriate. Kenaga suggested that residues on insects can be estimated from residue data for plants with similar surface area-to-mass ratios as the insects in question. He indicated that for small insects the residue data available for forage are relevant and for large insects the data for seeds and pods are pertinent. Therefore the revised estimates for forage and seeds and pods can be used for small and large insects, respectively.

[Note : These revised residues estimates should also be used for avian assessments.]

Granular Formulations

Exposure estimates for granular formulations used for mammals should be similar to those used for avian species: mg ai/ft^2. The question may be asked about the use of this index for mammals because of the obsession some have with avian species and granular exposure through grit consumption. Certainly for avian species this can be a major route of exposure of some granular products for some avian species. However, this index was designed or at least it was assumed to account for other routes of exposure, dermal, dietary, etc. Not all granular products are on carriers that mimic grit. Some are on corn cob particles that resemble seeds or other food sources. Also, granules may adhere to food sources of wild mammals such as insects, seeds or forage or dissipate from the granule and contaminate food sources. [This also applies to avian species in the use of this index.] **Therefore this index is appropriate for mammalian species as well as avian species.**

The following summarizes the methods used to calculate mg ai/ft^2:

$$\text{ai (mg)/ft}^2 = \frac{\text{(oz product per 1000 ft of row * \% ai) * 28,349 mg/oz}}{\text{(1000 ft * width of band or furrow (ft))}}$$

Exposed ai (mg)/ft^2 = ai (mg)/ft^2 * % unincorporated

The following format is recommended to present the results of these calculations:

Use/appl method	Formu-lation	Use Rate oz/1000ft row	width of applica-tion(ft)	%unicorp-orated	ai mg/ft^2	ai mg/ft^2exposed
Corn Banded at plant	20 G	6	0.6	15	56.70	8.51
Corn Infurrow at plant	20 G	6	0.1	1	340.2	3.40

If the label does not give application rate in weight of product per feet of row but gives application rates on a per acre basis, to estimate mg ai/ft^2 the amount of the acre that is treated must be calculated using row spacing and band width. The following outlines these calculations:

$$\text{Equivalent lb/acre rate on band or in-furrow} = \frac{\text{Label Application Rate (lb product/acre)}}{\text{Linear ft/acre (based on row spacing)}} \times \frac{\text{43,560 sq ft/acre}}{\text{Treated Row Width (ft)}}$$

where: a) Linear ft/acre is determined by dividing 43,560 sq ft/acre by row spacing; and

b) Treated row width value is determined to be 0.4-0.6 ft band (5-7" as per label) or 0.1 ft in-furrow, drill, and topical applications.

$$\text{mg ai/ft}^2 = \frac{\text{(Eq. lb/acre rate X 453,584mg/lb X \% ai X (\% unincorp.)}}{43,560}$$

The following format is recommended to present the results of these calculations:

Use/appl. method	Formula-tion	Use Rate lbs product/A	Row Spacing	width of appl.	% unicorp.	Linear ft/A	Treated area (ft^2)	equ.lb /acre	mg ai/ft^2 exposed
Corn Banded at plant	20 G	6.5	2.5	0.6	15	17,424.00	10,454.40	26.14	8.17
Corn Infurrow at plant	20 G	6.5	2.5	0.1	1	17,424.00	1,742.40	161.17	3.36

Risk Quotients

Risk quotients for mammalian species are calculated in the same manner as for avian species:

exposure/toxicity = RQ or

$$\text{mg ai/ft}^2/\text{LD}_{50_A} \times \text{Body wt.} = \text{RQ}$$

Non-Granular Acute Risk Quotients

The difference between avian and mammal calculations of RQ values is that in most cases for non-granular formulations the toxicity values need to be converted to equivalent units of estimated exposure concentrations. This is necessary because the mammalian dietary tests, in most cases, are not adaptable for calculating risk quotients for comparison to concern levels defined from median lethal concentrations (LC_{50}'s).The human and domestic animal sub-chronic tests are designed to permit the estimation of the no-observed-effect level not median lethal concentrations (LC_{50}'s). Due to this, risk quotients for wild mammalian species in general are based on the LC_{50} values calculated from LD_{50} values. LD_{50} values are reported in the standard units mg/kg, and therefore must be transformed into equivalent units for comparison to EEC's from Kenaga.

Its important to realize that the standard being used is not a typical LC_{50}, but a specified quantity of food which can be expected to be consumed in a day for which residues equal a single acute dose or LD_{50}. McCann et al. (1980) compared rat LC_{50} values with published rat single LD_{50} values and with LC_{50} values calculated from the published rat single-dose oral values. The data showed that the LD_{50} and LC_{50} values for rats cannot be used interchangeably and that the LC_{50} values calculated from the LD_{50} values are

generally not toxicologically equivalent to the LC_{50} values from the study. Kenaga (1977) made similar observations about avian toxicity tests. However, while not toxicologically equivalent both provide an estimate of an amount of chemical when exposed, the LC_{50}, or when ingested, the LD_{50}, that will cause adverse effects. The strengths and weaknesses of each can be discussed for use in indices to predict hazard, but because of the absence of a dietary study that estimates the LC_{50} for mammals, the LD_{50}, which is generally available and can be converted to a concentration on typical food sources, provides a tool to screen potential problems. Research has not been completed which evaluates the functional relationship between laboratory toxicity tests, either the LD_{50} or the LC_{50} tests, and field effects.

The calculation of the median lethal dose/ daily ingestion rate is where most of the differences between EEB evaluations appear with different species being selected and different food ingestion rates being used for similar species. The basis for the calculation however are the same. Equal sensitivity to the toxicant between mammalian species is assumed with the dose being adjusted in relation to food ingestion rates for the species selected. LOC's are then calculated for the species selected based on residues on their primary food source. The calculation used is:

concentration of toxicant in a days food = LD50 / % body wt
lethal to 50% of test population consumed

In years gone by (Urban and Cook 1986) this basic equation was used; however, the extrapolation was limited to the rat. The rat LD_{50} was converted to a rat mg/kg/day and expressed as ppm in the diet, based on the assumption that adult rats consume dry food equivalent to about 5% of their body weight. This value was then used for mammals in general for comparison with EEC's.

I've not found documentation of why or when we moved from the rat to rat extrapolation, to the rat to other mammal species. Urban and Cook (1986) do mention this approach; however, indicate it's not routinely used. The rationale would appear to be the low percent of body weight consumed by a rat in comparison to other small mammals. This low percent ingestion rate for the rat compared to other small mammals is a result of using dry weight equivalences and the relative larger body size of the rat. Nagy (1987) developed the following equation for calculating dry weight ingestion rates for rodents:

$$\text{dry weight ingestion rate(g/day)} = .621\ Wt^{0.564}(g)$$

Using this formula a 300 g rat consumes 15.5 g/day or 5.2 percent of its body weight as compared to a 20 g rodent which consumes 3.4 g/day or 17 percent of its body weight. This difference in ingestion rates on a dry weight basis results in approximately a 3

fold difference in the estimated concentration in food ingested per day lethal to 50% of exposed animals [20ppm compared to 6ppm based on an LD_{50} of 1 mg/kg]. However, if percentage of water in the food is accounted for, the difference is much larger. For example, grasses and forage are reported to be 75 to 85 percent water. Correcting for water content, assuming 80 %, the 20 g rodent would consume approximately 17 g/day or 85 percent of its body weight [3.4/x(g) = 1 - 0.80]. The estimated concentration in food ingested per day lethal to 50% of exposed animals would be 1.2 ppm or nearly 20 fold less than the 20 ppm rat value based on the rat dry weight extrapolation. If we assume the food source is seeds, which contain about 10% water, the estimated concentration in food ingested per day lethal to 50% of exposed animals would be 5.3 ppm. Therefore, the reasoning for using the rat to other species extrapolation over the rat to rat extrapolation is the latter does not account for differences in food ingestion rates and water content of food sources of wild mammals.

Accounting for the difference in food ingestion rates appears logical when you consider the standard it's based on. However, it should be noted that at first glance, the approach appears to be in contrast to avian evaluations where ingestion rates are not considered. While further consideration of this difference may be appropriate, let it suffice that the standard used in the avian and mammal evaluations is different.

As pointed out previously, in general, procedures used in EEB's assessments have been similar, with differences being in ingestion rates and species selected. Inasmuch as there is no "correct" species and the differences in conclusions are minor (as long as a reasonable range in ingestion rates were selected) the inconsistences are more perceived than real. However, to avoid further questions about EEB's approach some standardization seems appropriate.

In examining the theory underlying the mammal hazard evaluation the major variable is ingestion rate which varies between species depending on a number of factors. These include weight, age, sex, seasonal changes in ambient temperature, activity levels, reproductive activities, and the type of diet consumed. Each of these factors influences the metabolic rate which is a direct function of ingestion rate of the individual animal. However, it has been found that rough estimates of ingestion rates can be made for most species based on body weight and type of diet consumed. These two factors have been found to contribute more to differences in ingestion rates than variation between species. Similar size species consume relatively the same amount of food of similar types. Further, ingestion rates of different type foods are more a function of percent water content than other factors such as caloric content and digestibility. If we account for these two variables, body size and food type, gross estimates of ingestion rates can be made for a range of mammalian species, thus, allowing us to estimate toxic exposure levels.

Since the type of species is not a major contributing factor for estimating toxic exposure levels, modifying our mammal hazard assessment from a species-specific to one dealing with a range of body sizes related to food habits may help reduce the perceived inconsistencies of EEB's previous assessments. Also, the advantage to this approach is that it does not create the impression that we know more than we do about small mammal hazards. That is, these extrapolations are relative gross approximations based in most cases on one, or at most, a few laboratory tests on a single species. We do not know the toxicity of the chemical to the various species which are used as examples within the tables. This can and has been interpreted to suggest more of a specific concern for the species listed than the general screening tool it's intended to be for mammals.

Mammals can be separated into five general groups according to food habits: herbivores, insectivores, granivores, omnivores, and carnivores. For our purposes, at least in the initial screen, we can reduce it to three groups: herbivores, insectivores and granivores. The water content of the diet of herbivores and insectivores is approximately equal, 75% to 85%. Therefore ingestion rates for similarly sized individuals in these two groups should be similar, and differences in exposure levels can be accounted for when compared to EEC's. Omnivores will be covered by the range of these two groups (herbivores and insectivores) and since they may consume large percentages in their diet of seeds, forage, or insects, the extremes in EEC values are accounted for by the three groups. Carnivores, sense for the most part would be exposed through secondary exposure routes, and if biomagnification is not a concern, the higher exposure levels in the primary consumers should be protective of this group.

The other factor which needs to be considered is body size. Mammals range in size from a few grams to several hundred kilograms. If we limit our initial screen to the smaller mammals in the range of rodents and lagomorphes, the larger mammals which consume a lower percent of their body weight should be protected. In cases where potential problems are identified in the larger small mammals it may be appropriate to included larger body weights. Table two gives some average weights of representative small mammal species. While the selection of this particular range of weights is somewhat arbitrary, it seems that 15, 35 and 1,000 grams provides a reasonable range of body weights for the initial screen.

Using the previously cited equation developed by Nagy (1987) for dry weight ingestion rates, and assuming 10 percent water content for seeds and 80 percent for grasses, forage and insects based on information from the literature (table three), estimates of wet weight ingestion rates can be made:

$$\text{ingestion rate wet weight(g/day)} = .621\ Wt^{0.564}(g)/(1- \%\ \text{water content}$$

of food item)

Using the above equation the following ingestion rates for herbivores and granivores for the indicated body weights can be developed:

Body Weight	Ingestion Rates % of body weight	
	Herbivore/Insectivore	Granivore
15 g	95	21
35 g	66	15
1000 g	15	3

These ingestion rates then can be used to estimate the concentration in food ingested per day lethal to 50% of exposed animals using the following as previously suggested:

concentration of toxicant in a days food = LD50 / % body wt
lethal to 50% of test population consumed

RQ values then are calculated by dividing the estimated concentration in food ingested per day lethal to 50% of exposed animals by EEC's. The results of these calculations can be tabulated as follows:

Herbivores/Insectivores

Body weight (g)	%Body wt consumed	Rat LD_{50} mg/kg	Est. mg/kg/d in diet	EEC grass	EEC forage&sm insect	EEC lg. Insects	RQ grass	RQ forage &sm insect	RQ lg. Insect
15	95	1	1.05	240	135	15	228.57	128.57	14.29
35	66	1	1.52	240	135	15	157.89	88.82	9.87
1000	46	1	2.17	240	135	15	110.60	62.21	6.91

Granivores

Body weight (g)	% Body wt consumed	Rat LD_{50} mg/kg	Est. mg/kg/d in diet	EEC seeds	RQ
15	21	1	4.76	15	3.15
35	15	1	6.67	15	2.25
1000	3	1	33.33	15	0.45

Risk Quotient-Level of Concern Comparison

Body Weight (g)	RQ Herbivores	RQ Insectivores	RQ Granivores	LOC's
15	128-229	14-128	3.2	0.5
35	89-158	10-89	2.3	0.5
1000	62-111	7-62	0.5	0.5

The above three tables are recommended as a standard format. Before this exercise is initiated, however, some time may be saved be solving the following equation for the 15 g herbivore's Risk Quotient:

$$\text{short range grass EEC}/[\text{Rat } LD_{50}/.95] = RQ$$

If this RQ is found to be less than 0.5, no levels of concern will be exceeded. Therefore a summary statement in the mammal section stating this and concluding hazard is low to nontarget species should be adequate.

In all cases where results of the dietary test are available, the dietary values should be factored into the assessment to help put the hazard potential into perspective. It has been found in some instances that rats exposed to concentrations several times the calculated LC_{50} values for extended periods have survived with no apparent adverse effects. However, the following two factors should be noted: 1) the above calculated indices are based on a different standard; and 2) the ingestion rate of laboratory rats is relatively low. Judgement will be required to determine if differences between the sub-chronic and chronic test results override the acute test results. In instances where a clear conclusion cannot be made, the need for a wild mammal dietary test should be considered.

Granular Acute Risk Quotients

The RQ for granular formulations is: the exposure estimate divided by the median lethal dose to an individual. Again, I would recommend using a range of weights instead of individual species to avoid indicating we know more than we do about small mammal hazard. The variable which we try to account for in this index is the amount of toxicant lethal to different sizes of animals with sensitivity between the animals being equal. As for ingestion rates in evaluating non-granular products, the range of body weight selected is somewhat arbitrary, but for consistency with non-granular assessments the same weights can be used here: 15, 35, and 1000 grams. The following equation can be used to calculate RQ values for granular products:

RQ = mg ai/ft^2 /LD_{50} * body weight (kg)

The results of these calculations can be tabulated as follows:

Body Weight(g)	Rat LD_{50} mg/kg	mg ai/ft^2 exposed	RQ	LOC
15	1	8.5	567	.5
35	1	8.5	243	.5
1000	1	8.5	8.5	.5

Chronic Non-granular Risk Quotients

The calculation of chronic risk quotients to mammals uses the EEC's on mammalian food sources (table one) divided by the NOEL's from the mammalian reproduction study submitted in support of human toxicity data requirements. As with acute hazards, if mammals are less sensitive to chronic effects than avian species, it can be assumed that the avian assessment will be protective of mammals. If a potential chronic effect is suggested for avian species, or if birds are more sensitive, then further consideration of mammal species seems germane to provide a more complete picture of potential impacts to non-targets.

The initial calculations are straight forward, that is:

RQ = EEC/NOEL

The following format can be used to summarize this information:

Food source	Use rate lbs/A	EEC (ppm)	NOEL (ppm)	RQ	LOC
Short grass	1	240	250	0.96	1
Long grass	1	110	250	0.44	1
Broadleaf plants	1	135	250	0.54	1
Fruit & seeds	1	15	250	0.06	1

If this initial screen suggests a potential chronic hazard, further evaluation may be necessary. Several points need to be considered and a great deal of judgement is required. These points include:

* What effects and the magnitude of effects were observed in the mammal reproductive study? In some instances the effect or magnitude of the effect found in these studies may not be of biological significance to wildlife species. A copy of the study may be required to evaluate the significance of the observed effect.

* What are the exposure estimates, their extent and duration? In most instances, field dissipation data, especially for residues in or on wildlife food items will be scant or not available. Such data usually concern residues in soil, water, and in or on crop parts intended for human consumption.

* What is the potential for bioaccumulation?

* What are the residue levels in relation to acute hazard. If there is a potential for acute hazard or initial residues are approaching levels which may suggest acute hazard, this may give support to potential chronic exposure levels occurring.

Chronic Granular Risk Quotients

Chronic risk quotients for granular products have not been previously defined for either mammals or avian species. While, I'm not sure as to why, some of the reasons may be related to the toxicity of most of these chemicals to mammalian species. One of the major reasons these chemicals are formulated as granular products is to reduce potential human exposure because of the liquid formulations' toxicity to mammals.

In a few assessments, of which I'm aware, chronic risk quotients have been calculated using "Kenaga" EEC estimates and LC_{50} values assuming a foliar application at label rates. Also, residues in the top few inches of soil have been entertained by a few to estimate potential exposure levels for evaluation of the potential chronic hazard from these products. In the development of this paper I explored several other methods which included: assuming that residues from a single application would be similar to those from a foliar application, but layered in the soil profile such that residues are diluted as if 10 to 20 partial applications were made to each layer depending on the depth of incorporation. A defined percentage of Kenaga residues based on the depth of incorporation then could be used to develop RQ values.

Each of these methods could be used to estimate exposure indices and are based on application rates which could be argued to correlate to exposure to some extent. However research, as is the case with other indices used in pesticide evaluations, is not available to define this correlation. Further, each of these estimates is based on a somewhat different reference point than the acute index for granular products, LD_{50}'s/ft^2.

While the LD_{50}'s/ft^2 also has limited data on its functional relationship to field effects, it has been used as the standard for granular products as a basis for estimating potential impacts. However, LD_{50}'s, or maybe more appropriately in the context of chronic hazard, mg's/ft^2 have not been extended to chronic exposure estimates. This appears to be due to the difference in units in exposure and laboratory effects, mg/kg and ppm. This can be overcome by taking a similar approach as used for the non-granular exposure estimates for mammals, but in this case the laboratory chronic value is converted to equivalent units as the exposure estimate, mg's/ft^2. Since, the route of exposure in the mammalian laboratory reproduction study is the diet, as is the case for the avian reproduction study, using similar assumptions in extrapolating to other species seems appropriate. That is, the benchmark for effects is the concentration in the diet of the indicator species and it's assumed that other species will respond similarly at similar concentrations in the diet.

The mammalian chronic RQ then can be defined as:

$$RQ = mg's/ft^2/[NOEL\ (mg/kg/day) * body\ weight\ (kg)]$$

The average adult laboratory rat weighs approximately 400 g and consumes approximately five percent of its body weight a day. Therefore to calculate the mammalian chronic RQ value, the NOEL, usually reported in ppm, from the mammal reproduction study is converted to mg/kg/day using the following equation:

$$mg/kg/day = \%\ body\ weight\ consumed\ X\ residues\ in\ diet\ (ppm)$$

Then the RQ is calculated using the previous equation. The results

of these calculations can be tabulated as follows:

Rat Body Weight(kg)	Rat NOEL PPM	Rat NOEL mg/kg/day	mg ai/ft^2 exposed	RQ	LOC
.4	250	12.50	8.5	1.70	1

If the chronic RQ exceeds the LOC, further evaluation, that considers environmental chemistry, the magnitude the RQ exceeds the LOC, etc. is needed to put potential chronic hazard in perspective.

Table one. Estimated EEC's on Wildlife Food Sources

Plant, organ category	Residue(ppm) following 1 lb a.i./acre application
Short grass	240
Long grass	110
Broadleaf plants[1] leaves and leafy crops forage e.g.,alfalfa	135
Fruits[2] Pods and seeds	15

1. Residue estimate applicable to small insects.
2. Residue estimate applicable to large insects.

Table two.

Order/Family	Species	Food Habits	Body Wt
Insectivora/ Soricidae	Short-tailed Shrew *(Blarina brevicauda)*	Insectivore	15-22 g
	Least Shrew *(Cryptotis parva)*	Insectivore	4-7 g
	Masked Shrew *(Sorex cinereus)*	Insectivore	3-6 g
Rodentia/ Muridae	Deer Mouse *(Peromyscus maniculatus)*	Granivore	15-35 g
	White-footed Mouse *(Peromyscus leucopus)*	Herbivore	15-35 g
Rodentia/ Muridae	Prairie Vole *(Microtus ochrogaster)*	Herbivore	30-40 g
	Meadow Vole *(Microtus pennsylvanicus)*	Herbivore	20-40 g
Lagomorpha/ Leporiae	Eastern Cottontail *(Sylvilagus floridanus)*	Herbivore	.7-1.8 kg
	White-tailed jackrabbit (*Lepus townsendii*)	Herbivore	2.2-4.5 kg

Table three.

Type of Food	% H_2O
Invertebrates	
earthworms	84
grasshoppers	69
Beetles	61
mean	71
Plants	
young grasses	70-88
mature dry grasses	7-10
Leaves and leafy crops	85
Forage (alfalfa, clover)	85
seeds	10
fruit	77